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GIS-Based System for Surface Water Risk Assessment of Agricultural Chemicals. 1. Methodological Approach

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A methodology to develop a GIS-based system for the surface water risk assessment of agricultural chemicals is described. It is based on the integration of relational and spatial databases, GIS incorporating raster and vector, mass balance models, and pesticide risks indicators. Surface water pollution was modeled by taking into account two main processes: the load due to drift and the load due to a rainfall−runoff event. The former is immediately consequent to pesticide application; the second occurs a short period afterward. Thus two distinct PEC (predicted environmental concentration) values were estimated, differing in time. A pilot approach was applied to the herbicide alachlor on corn in Lombardia region (northern Italy) and represents the first stage of a wider project. Although the resultant alachlor PEC and risk maps represent a static image of a worst-case simulation, the main objective was to provide information for the territory with respect to relative risks at the watershed level, which is important in managing risks to the aquatic environment. The driving forces and spatial variability of the above-mentioned processes were investigated to improve knowledge about the territory and to indicate the need for more detailed site-specific studies.

Introduction

Nonpoint source pollutants (NPSP) are a problem of increasing concern all over the world due to their presence in water and soil and their potential adverse effects on human health and the environment. NPSP are generally detected in medium or low concentrations in environmental compartments, but their behavior makes them widely distributed in nature and makes it impossible to trace a specific and localized source. A recent Water Quality Inventory in the United States reports that agricultural activities (e.g., fertilizing, irrigation, pesticide application, and others) are the main sources of NPSP (*1*).

Among these substances, pesticides are intentionally introduced into the agricultural ecosystem to control pests or weeds. Pesticide applications save up to 40% of crop losses, and so they are generally accepted as having a valuable function, but their movement toward environmental compartments, such as surface waters, potentially extend effects on nontarget ecosystems beyond local boundaries and may cause a reduction of the possible uses of water resources, first of all potability (*2*).

For a site-specific assessment of the impact of pesticide on surface water and for proper land use management, environmental fate and ecotoxicological risks should be assessed taking into account intrinsic ecotoxicological properties of the chemicals and characteristics of the territory (morphometry, land use, soil characteristics). This approach would allow for guidelines to be provided for decision-making authorities on a suitable watershed scale.

To identify the driving forces of the processes involved in pesticide movement, a multidisciplinary approach is needed, and a complex of correlated factors must be considered. Several fate and transport models of varying complexity and predictive capability may be used to evaluate PEC (predicted environmental concentration) in surface waters. In particular, the FOCUS working group (*3*) is now selecting more suitable models for predicting pesticide exposure to surface water.

Exposure and effect parameters can be combined in risk indicators, developed as comparative tools for ranking chemicals, based on standard environmental scenarios unrelated to the real characteristics of the territory (Supporting Information, NB1). Nevertheless, if transferred to a site-specific environmental condition with known characteristics (e.g., land use, soil properties, meteorological data), risk indicators may represent useful tools in assessing environmental risk on the territory.

The integration of models and geographical information systems (GIS) is effective in addressing the problem of spatial and temporal variability of the different parameters involved in environmental processes and in producing risk cartography. Currently the link between models and GIS is implemented by inputting data required by the model within a GIS, transferring them automatically to the external model (*4*), re-importing the model output into the GIS, and recreating a georeferenced context to analyze and display the results.

In the present work a GIS-based methodology is developed in which the model is directly implemented into a GIS, for surface water risk assessment of agricultural chemicals. A pilot approach was developed and applied to the herbicide alachlor in the Lombardia Region (northern Italy).

The present paper represents the first step of a wide research project, and it is aimed at a description of the methodology. Further steps are under development (*5*), including the application to several active ingredients (*6*), for a better calibration of the procedure and the experimental validation of exposure prediction and risk assessment.

Materials and Methods

The methodology is based on the integration of a relational database, a spatial database, GIS, both raster and vector technology, and mass balance evaluating models. All the data are stored or translated to the Italian coordinate system Gauss Boaga, and the raster format is characterized by a 200-m pixel size, which is the most suitable distance for a realistic environment description with the number and typologies of data available (Supporting Information, Tables 1 and 2).

Collected data included land use, soil properties, rainfall, topography, pesticide use, phenological stage of crops, and hydrographic network. Data were stored and organized into two different databases. The first, built inside the GIS, consists

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of geographical data. The second deals with environmentally relevant pesticide properties, namely, the physical and chemical properties governing molecule environmental behavior and the ecotoxicological end points (acute toxicity) for nontarget organisms representative of surface water ecosystems (*7*).

Since the 1970s, the agricultural use of pesticides, although less persistent than former ones, has increased dramatically. Many of these pesticides are more water-soluble than the previously used organochlorine compounds, increasing the likelihood of transport to surface waters. Only a few pesticides are strongly adsorbed by soil particles, and dissolved runoff losses generally exceed solid-phase losses, especially for herbicides (*8*). For that reason, sediment transport was excluded from the current calculation.

Surface water pollution by agricultural chemicals depends on two main processes, the load due to drift and the load due to a rainfall−runoff event. The former is immediately consequent to pesticide application; the second occurs a short period afterward. By consequence, two distinct PEC values were estimated for two different dates.

Volatilization losses are frequently overlooked in pesticide fate modeling. Recent studies demonstrated that high volatilization of herbicides such as alachlor and triazines may occur on dry soil, while losses are not relevant after a rain event (*9*). Therefore, in the scenario adopted in this work, volatilization was assumed as negligible. Groundwater may also contribute to surface water pollution; nevertheless, the different transport patterns and the long time needed require different study approaches.

PECs are estimated by the simple deterministic modeling approach described below. TERs (toxicity exposure ratios) for selected groups of organisms, representative of the surface water ecosystem (namely, algae, *Daphnia*, and fish) were calculated using PEC values and toxicological end points.

TER values are integrated in the risk index PRISW-1 (Short-Term Pesticide Risk Index for Surface Water; Supporting Information, NB1 and Tables 3 and 4) according to a recently developed method that evaluates risk immediately after pesticide application (*10*). With the proposed methodology, two concentration peaks can be estimated, one for physical drift and one for runoff, in the water body segment that directly receives polluted water from a watershed portion. This means that phenomena occurring within the riverine flux (namely, advection, dispersion, and partitioning with sediment of the river bed) were not considered in this methodology. The resultant PEC maps provide a static and instantaneous picture, referring to each emission event (drift or runoff) and to each portion of watershed without taking into account the dynamic processes of the hydrological network. These aspects are the focus of the ongoing second part of the research project.

The GIS software used are ArcView 3.1 (*11*) for vector operation and ILWIS (Integrated Land and Water Information System) version 2.23 (*12*) for raster operation; the methodology also includes an exchange protocol between the two mentioned GIS. To develop the relational database concerning pesticide properties, MS Access 8.0 was used.

Description of the Area. The Lombardia Region (Figure 1) is located in the northern part of Italy and has an extension of 23859 km^2, 47% of which is plain, 12% is hilly land, 41% consists of mountains, mostly in the northern part of the region with a small portion in the south, beyond the Po River. Most of the flat areas extend on to the central part of the region (Po Valley) while a minor percentage forms the bottom of alpine valleys. Another characteristic is the relevant presence of water bodies, among which are 77 with a considerable watershed and several rivers with water flow higher than 150 m^3/s, such as the Po and its main tributaries (Ticino, Adda, Oglio, and Mincio).

The climatic conditions of the flat region can be described as temperate continental with an average temperature, referring to the most cultivated area, that spatially ranges between 12.5 and 14 °C. January is the coldest month (0.2−3.9 °C), and July is the hottest (23.3−25 °C) (*13*). Total rainfall is about 900−1000 mm/yr (*14*). Rainfall events are not uniformly distributed during the seasons, but two peaks are usually registered in the monthly data patterns. The higher rainfall occours during spring and the other in autumn.

The climatic conditions, the relative abundance of water, and the flat landscape morphology support intensive agriculture. In particular, corn is the most relevant crop in the region (Supporting Information, Figure 1).

In the area studied, herbicides are used at the beginning of the growing season, corresponding to the above-mentioned rainfall peaks in temperate regions. Alachlor is the most commonly applied herbicide on corn (*15*), and it is was chosen as a representative example for a surface water risk map of the Lombardia Region.

Description of Modeling Scenario. The PEC was estimated by means of a deterministic approach. To fully describe the processes, many input data and their spatial and temporal variability have to be taken into account. In any case, the methodology, working with a worst case scenario, permit us to obviate an eventual lack of information, in particular concerning the timing and the location of pesticide treatments. This procedure allowed to apply a common scenario to all basins in order to obtain results with comparable meaning. The main default worst-case assumptions are that all the farmers make the chemical application on the same day with the highest rate of application on the entire surface of the crop (corn) and that the application event is followed by the rainiest day of the potential application period. Alachlor is a herbicide used against gramineous weeds and is usually distributed with a single application on corn during its preemergence period, which in Lombardia may occur from the last ten days of March to the second ten days of May, according to precocity class and local conditions (*16*).

Daily meteorological data for the entire region were stored in the geographical database for the period 1988−1997. The year 1996 represents a particularly rainy year on the plain and within the alachlor application period, April 2 was identified as the rainiest day with an average of 15.1 mm, a minimum of 1.3 mm in the Alta Valtellina zone (close to the north−east boundary of the region), and a maximum of 34.8 mm in the flat area close to the Po River. At this date, the worst case PEC due to runoff is calculated as a meaningful example of a potential adverse environmental occurrence. Since farmers generally are informed of weather forecasts to avoid pesticide applications too close to rain events, the alachlor application was assumed to occur March 30; therefore, this is the date for the estimate of a worst case PEC due to drift.

Drift Submodel. The application of agricultural chemicals results in the interception of spray drift in ditch water flowing close to treated fields (*17*). Through the connected hydrological network, polluted water may reach main natural water bodies, and since on flat regions artificial hydrological networks may easily overcome natural basin boundaries, an impact should be expected also for water bodies running through territories where there are no treated areas. Drift losses for different crops were assumed according to Ganzelmeier (*18*).

To achieve an estimate of drift contribution at watershed scale, the lost fraction of AI (% *D*) was calculated according to the scheme described in Figure 2a. Each crop was classified into one of the Ganzelmeier crop classes (*18*). The studied area was divided by a grid with 16 km^2 square cells. For every cell, an index of drainage density was calculated by summing

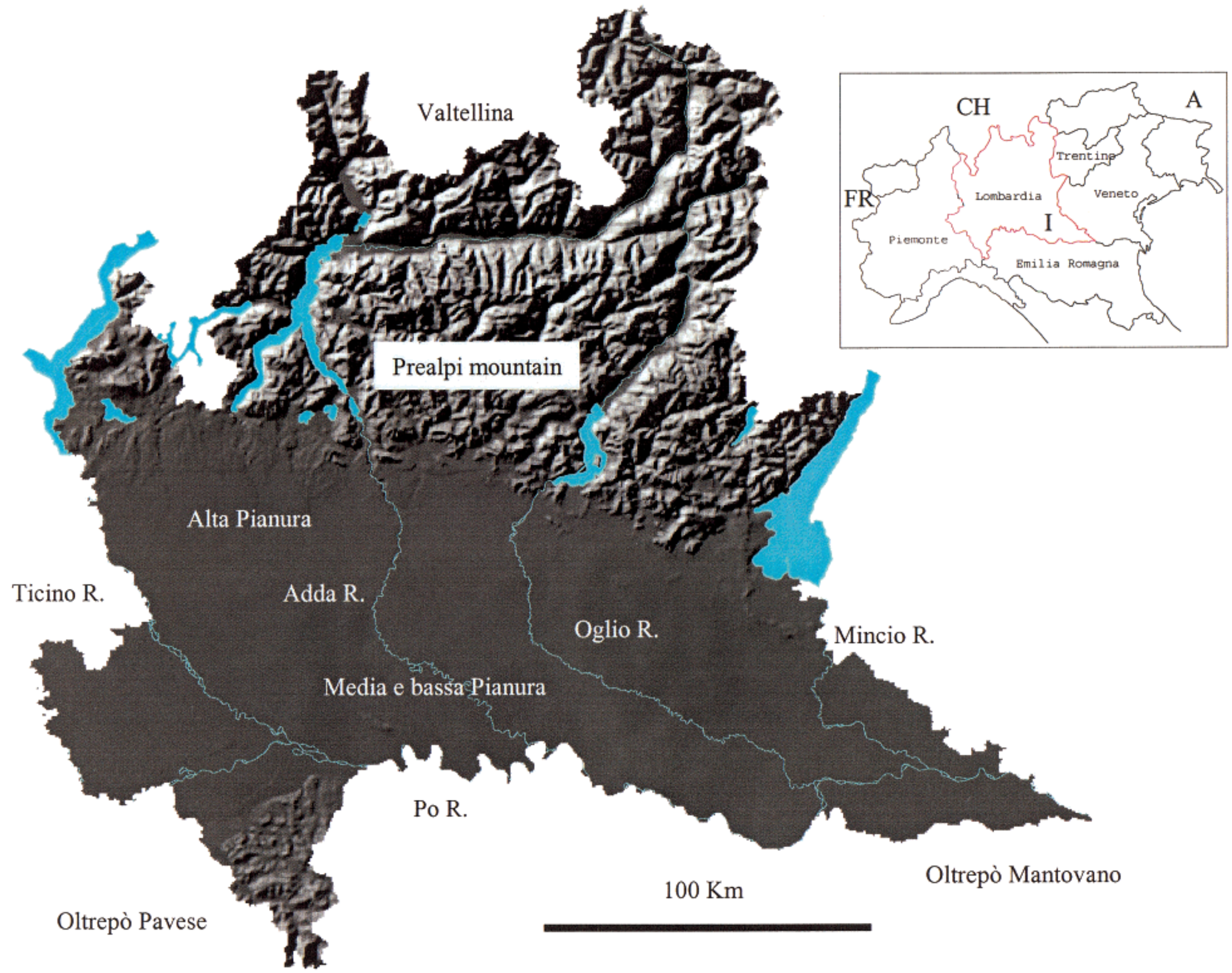

FIGURE 1. Studied area: Lombardia Region located in the northern part of Italy has an extension of 23859 km^2; relevant is the presence of water bodies with a considerable watershed and rivers with remarkable water flow, such as Po and its main tributaries, Ticino, Adda, Oglio, and Mincio.

the total length of the ditches and of the secondary natural tributaries.

Irrigation ditches do not usually generate inflow into natural water bodies, unlike drainage ditches: information about the ditch function (irrigation or drainage) was available only for some representative subareas (*19*). Above the spring line defined as the line where the piezometric surface reaches the topographic surface, 10% of ditches play a drainage function while downstream this percentage increases to 50%. According to this observation, for grid cells above the spring line, only 10% of the total length of the ditches was taken into account, 50% for downstream (Supporting Information, NB2).

Drainage density index (DDI) was calculated for each grid cell as follows:

$$\text{DDI (dimensionless)} = L\ (\text{m}) \times W\ (\text{m})/A\ (\text{m}^2) \quad (1)$$

where L is the total length of drainage ditches in the considered cell, W is the width of drainage ditches, an average W of 1 m was assumed for every ditch, and A is the surface area of the grid cell (16 km^2).

A DDI range from 10^{-7} to 10^{-2} was assumed. The maximum corresponds to a 100 m ditch for each hectare of the field. The minimum corresponds to the order of magnitude of the lowest calculated values in the study area (0.001 m for each hectare of field).

To calculate drainage density, the GIS layers of the ditches and of the secondary natural tributaries were elaborated from Arc/Info coverage of 1:10000 cartography (*20*). The spring line was identified georeferencing 166 resurgence sites described by Pisoni and Valle (*21*).

The DDI map was developed by dividing the study area into squares of 200 m (4 ha). A linear relationship was assumed to calculate the percentage of applied AI lost by drift (% D) in function of drainage density:

$$\%\ D = k \times \text{DDI} \times 100 \quad (2)$$

where kD is a proportionality factor depending on crop and crop stage according to the 95th percentile drift values reported by Ganzelmeier (*18*). For example, for arable crops, the early stage k value is calculated as follows:

$$k = \mathit{0.04} \times \mathbf{1} + \mathit{0.016} \times \mathbf{1} + \mathit{0.009} \times \mathbf{1} + \mathit{0.006} \times \mathbf{1} + \mathit{0.005} \times \mathbf{1} + \mathit{0.003} \times \mathbf{5} + \mathit{0.002} \times \mathbf{5} + \mathit{0.001} \times \mathbf{15} = 0.116 \quad (3)$$

where the distances between sources and ditches (in meters) are written in bold and the fraction of active ingredients lost for each distance is written in italic.

Crop stage, early or late, may be evaluated on phenological calendars referring to local conditions (*19*). The % D map for corn was developed (Supporting Information, Figure 3).

Runoff Submodel. Runoff is one of the main processes of water pollution by pesticides (*22*). Runoff occurs if rainfall

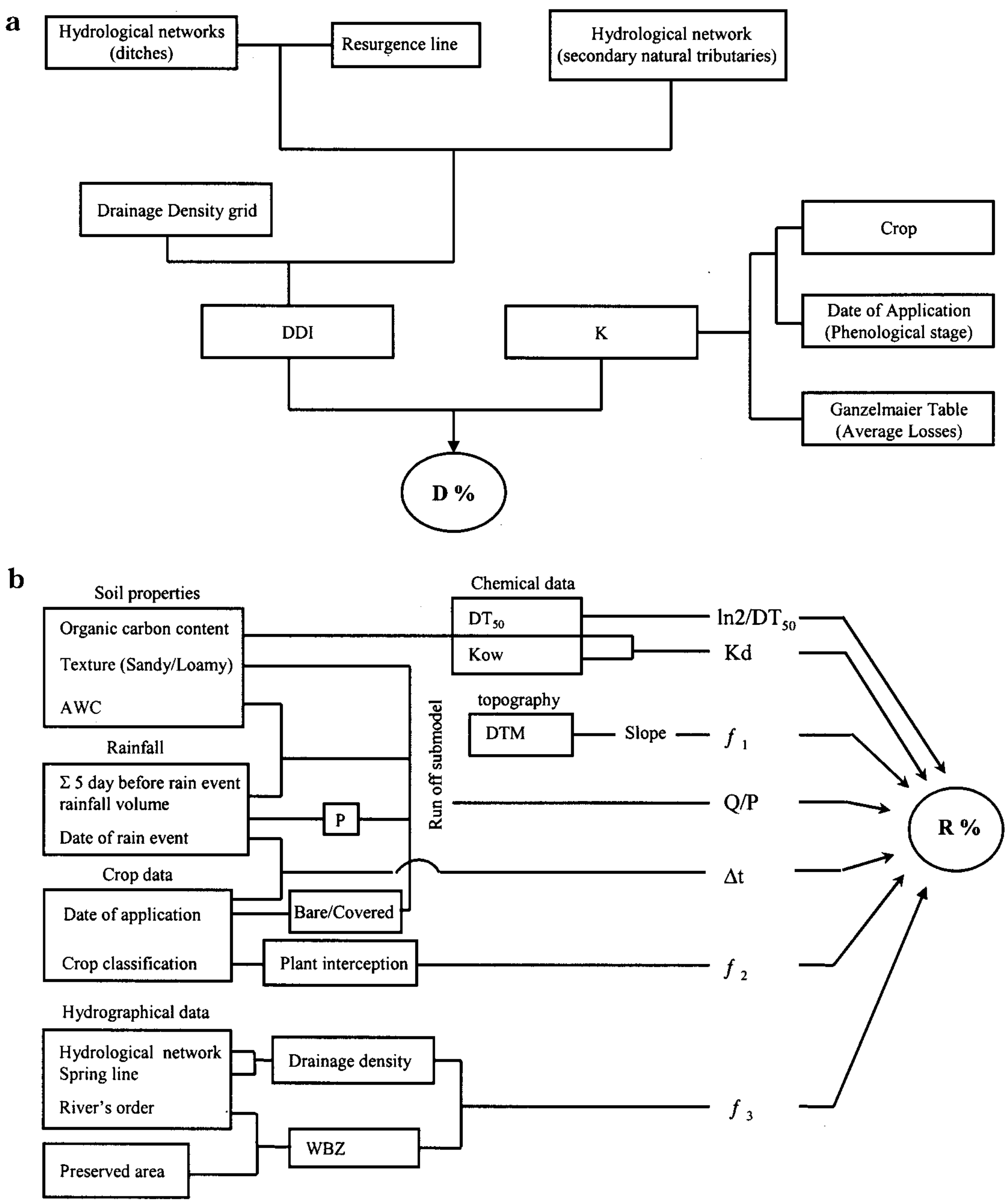

FIGURE 2. (a) Drift scheme to calculate the fraction of AI lost by drift (% *D*). DDI, Drainage density index calculated for each grid cell; the studied area was virtually divided. *k*, proportionality factor depending on crop and crop stage according to 95th percentile drift values reported by Ganzelmeier (*18*). (b) Runoff scheme to calculate the fraction of AI lost by runoff (% *R*) with the equation proposed by Gutsche and Rossberg (*23*). *Q*, runoff volume (mm); *P*, daily rainfall (mm); f_1, slope factor; f_2, plant interception factor; f_3, buffer zone factor; Δt, time between pesticide application and the first subsequent rain event; K_d, soil sorption coefficient

is sufficient to produce surface water flow (i.e., saturate soil porosity and overcome infiltration speed). Runoff load depends on geomorphological and pedological characteristics of the area, land surface conditions, and crop stage.

To calculate the fraction of AI lost by runoff (% *R*), the following equation proposed by Gutsche and Rossberg (*23*) was used:

$$\% R = \frac{Q}{P} f_1 f_2 f_3 \mathrm{e}^{(-\Delta t(\ln 2/DT_{50\mathrm{soil}}))} \times \frac{100}{1 + K_\mathrm{d}} \qquad (4)$$

where Q is the runoff volume (mm) calculated according to the model of Lutz (*24*) and Maniak (*25*), P is the daily rainfall (mm), f_1 is the slope factor, f_2 is the plant interception factor, f_3 is the buffer zone factor, Δt is the time between pesticide applications and the first subsequent rain event, and K_d is the soil sorption coefficient. (Details on each parameter estimated are provided in the Supporting Information, NB3.) The only modified parameter was the buffer zone factor, f_3. It was proportional to the width of the buffer zone (WBZ) in the original formula reported by Gutsche and Rossberg (*23*), but in the current methodology, we consider that runoff may also reach ditches or small tributaries of the rivers, which bypass buffer zones. So the denser the hydrological network is, the higher is the amount of pesticide reaching the river.

With the eq 4, % *R* was calculated at watershed level according to the scheme described in Figure 2b and the resultant map was developed (Supporting Information, Figure 4).

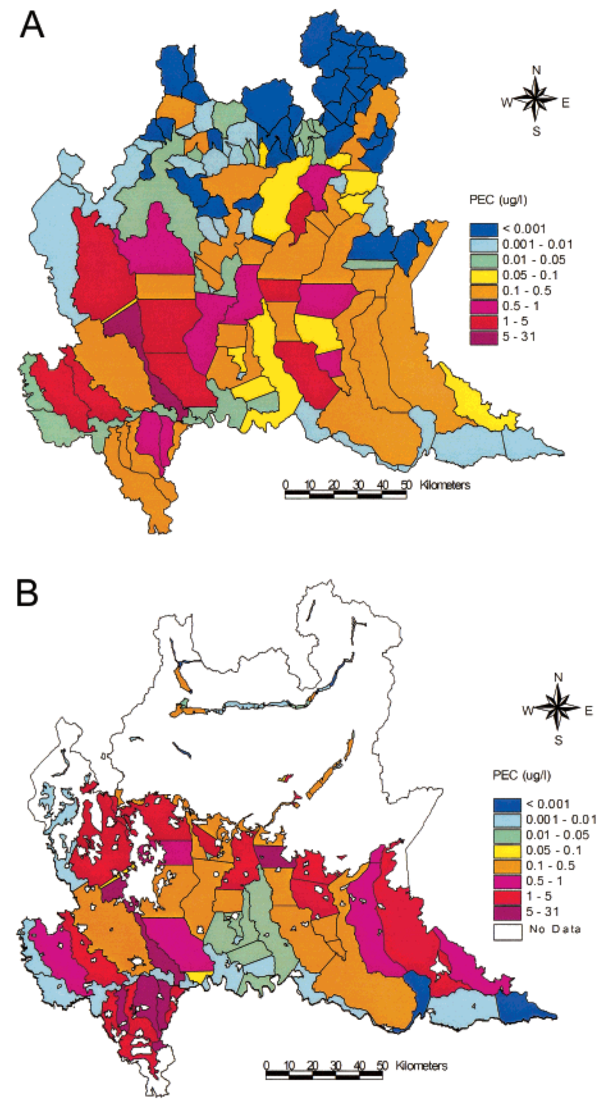

FIGURE 3. PEC of alachlor due to drift (a) and to runoff (b). Concentrations are reported for each river subbasin; for graphical reasons, rivers are not drawn. Runoff is calculated only for the 53% of the study area because of lack of pedological data. Either slope or rainfall of the considered day play a key role in determining the spatial variability of runoff PEC.

In present work, simple of environmental distribution models were used because more complex models, such as PRZM and Macro (*3*), need a huge amount of information to describe the complexity of such a wide territory, and it could make the methodology probably inapplicable. Actually, the difficulty in obtaining all the data required entails that unknown parameters are substituted by generic default values, so results do not fit with the peculiar landscape of the region analyzed.

Receiving Water Body Flow Estimate. A water flow georeferenced database was developed by digitizing and storing data records from 171 measuring stations (including river and ditches) of the public water quality sampling program established according to EU regulations (*26*). Among these, representative data of water flow were extracted for most of the 77 rivers considered. For rivers without measuring stations, a single default flow value was attributed in function of the order, watershed area, and upstream hydrological network. Measuring stations were digitized on hydrological network layer. If more than one station was present on the same river, this was divided into segments, and its drainage basin was divided into virtual portions, drawing a line perpendicular to the river segment in correspondence of the station. Hence, Lombardia was divided into 130 basins or portions of basin with corresponding water flow values at their outlet.

PEC Calculation. Two distinct PECs were estimated for drift and runoff, respectively. The % *D* or % *R* values in the respective maps were multiplied by the Application Rate (AR) of the considered AI on the considered crop and by the fraction of land surface covered with the crop as reported in a crop distribution (CD) layer. The results are the maps of the drift or runoff lost masses for each pixel of the raster grid. The equations are

$$\mathrm{DRIFT\ (mg)} = \%\, D \times [\mathrm{AR\ (mg/m^2)} \times \mathrm{CD\ (m^2)}]/100 \quad (5)$$

$$\mathrm{RUNOFF\ (mg)} = \%\, R \times [\mathrm{AR\ (mg/m^2)} \times \mathrm{CD\ (m^2)} - \mathrm{DRIFT}]/100 \quad (6)$$

where % *D* is the percentage of active ingredient lost by drift, % *R* is the percentage of active ingredient lost by runoff, AR is the application rate, and CD is the crop distribution. The pixels are distributed into the 130 basins or portions of basin. The total mass lost from each basin or portion of basin was calculated by means of the GIS summarizing operation. The ratio of the result (SUM_DRIFT or SUM_RUNOFF) by the water flow value (m^3/day) provides the PEC of the given day for each river segment as follows:

$$\mathrm{PEC\ (RUNOFF\ or\ DRIFT)\ (\mu g/L)} = \mathrm{SUM\_RUNOFF\ (or\ SUM\_DRIFT)\ (mg)}/[\mathrm{water\ flow\ (m^3/s)} \times 86400\ \mathrm{(s/day)}] \quad (7)$$

TER Calculation and PRISW-1 Elaboration. Toxicological data on aquatic organisms were taken from the database of the active ingredients (*7*). In particular for alachlor the considered values were as follows: *Daphnia* (48 h) EC_{50} 10 000 μg/L, fish (96 h) LC_{50} 1800 μg/L, and algae (96 h) EC_{50} 60 μg/L. The values were divided by the PEC of each subbasin producing six TER maps, three for drift and three for runoff, one for each of the nontarget organisms representative of the surface water ecosystem. PRISW-1 was applied to produce the risk maps for these two processes.

Results and Discussion

Drift PEC Map. The drift PEC map (Figure 3a) shows values ranging from 0 to 30.9 μg/L. Drift contributions are zero for the mountainous alpine zones. The maximum value (30.9 μg/L) is located in the southern part of the Olona River and is almost 1 order of magnitude higher in comparison with the second highest value (4.95 μg/L, Mella Basin). This value was due to an unusually low water flow (0.05 m^3/s) selected in the database.

The figures show good agreement with available experimental data according to De Snoo and de Wit (*27*), who reported that the average drift deposition in the ditch ranges from 0 to 1.1% at a wind speed of 4.5 m/s at different distances from the sprayed field (from 0 to 6 m buffer zone). Our results indicated that 27% of the studied area is characterized by the presence of river segments with PEC values included in the class $10^{-3}-10^{-2}$ μg/L and 25% in the class 0.1−1 μg/L. The high values in the western central zone correspond to densely corn cultivated areas but are even better correlated to the low water flow of rivers and the high drainage density. In the eastern part of Italy, values are lower notwithstanding wide corn surfaces and a mean drainage density because of the dilution into higher water flow.

Runoff PEC Map. The runoff PEC was calculated only for the 53% of the study area because of lack of pedological data,

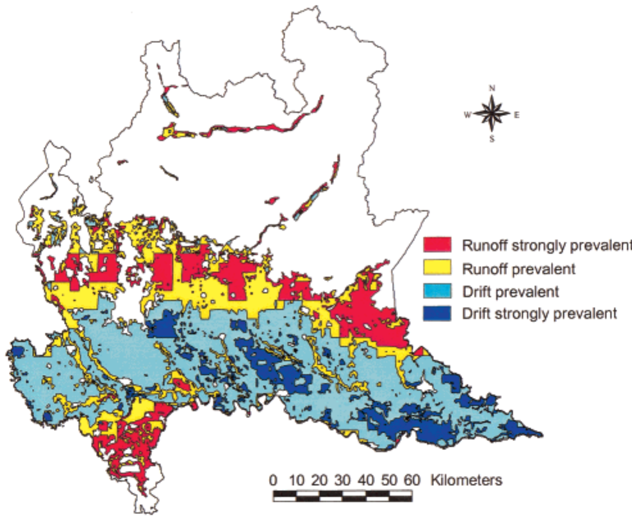

FIGURE 4. Runoff vs drift ratio of alachlor calculated, for the same area, for each pixel to understand the spatial distribution of the relative importance in determining surface water pollution phenomenon of the two considered processes: runoff is dominant in the northern part of the flat region and in the mountains while drift prevails in the central and southern parts of the flat area.

particularly in the mountains and urban areas. The runoff PEC map (Figure 3b) is characterized by a range from 0 to 11.5 μg/L. The maximum corresponds again to the southern part of the Olona River and is probably overestimated due to the same reason mentioned above.

Most of the figures are in the class from 0 to 0.01 μg/L, which represent 34.9% of river segments, and in the class from 0.1 to 1 μg/L (25.5% of river segments). These values are in agreement with experimental measures produced in Lombardia to detect what is known as the "spring flush effect" (*28*) and with prediction by site-specific model application (*29*). About the validation topic, see NB4 in Supporting Information. These values were distributed in the middle plain and in the lowlands where corn is widely cultivated but the soil slope is negligible. The figures close to zero are also determined by high water flow as in the case of the Po River. The higher classes can be related to areas where the slope increases although corn is not very widespread, such as Oltrepo Pavese or in the morainic zone. Either slope or rainfalls of the considered day (more than 25 mm in most of the southern part of the study area) play a key role in determining the runoff PEC spatial variability.

Comparison between Drift and Runoff. To evaluate the spatial distribution of the relative importance in determining surface water pollution phenomenon of the two considered processes (Supporting Information, Figure 5a,b), the ratio between drift and runoff of alachlor was calculated for each pixel. The results are shown in Figure 4, according to the following four classes: runoff strongly prevalent (ratio from 0 to 0.10), runoff prevalent (ratio from 0.11 to 0.99), drift prevalent (ratio from 1.01 to 10), and drift strongly prevalent (higher than 10).

The study area is clearly divided into horizontal zones: runoff is dominant in the northern part of the flat region and in the mountains while drift prevails in the central and southern parts of the flat area. This agrees with distribution of the values that enhance both processes (i.e., slope for runoff and DDI for drift) but it also relates to the spatial distribution of the rainfall amount. Therefore, considering a different rain event may change the relative extension of the zones, particularly the intermediate ones.

Risk Maps for Alachlor. Risk assessment for surface water was performed by means of the risk index PRISW-1 that

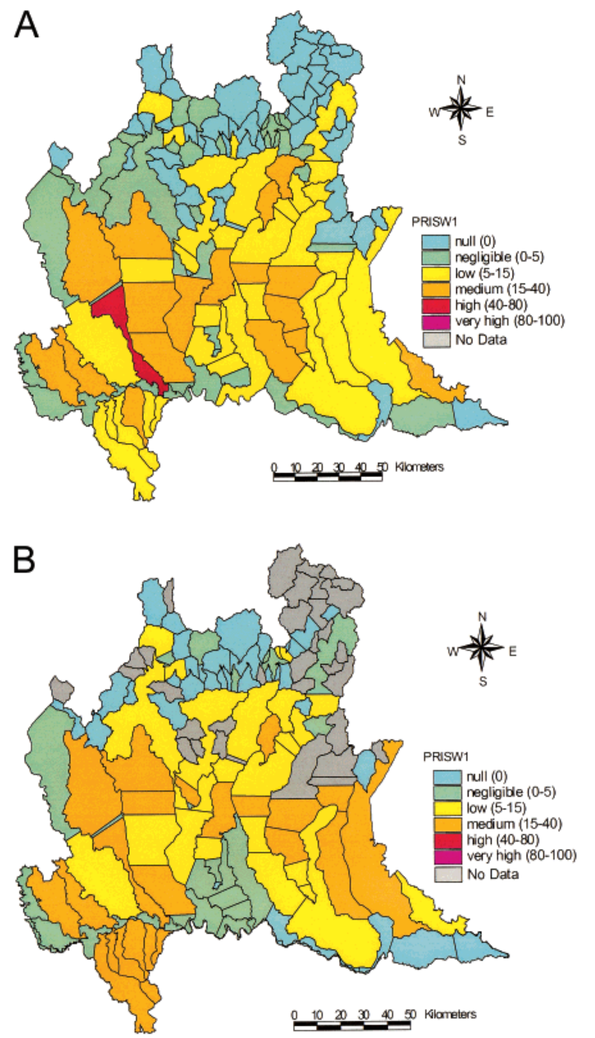

FIGURE 5. PRISW1 alachlor due to drift (a) for the 130 portions of basins on March 30, 1996. PRISW1 alachlor due to runoff (b) refers to the rain event of April 2, 1996.

integrates the TER values calculated for all three nontarget organisms. The individual TER calculated, according to EU regulatory triggers (*30*), do not indicate a risk for nontarget organisms, except for some particular situations (Supporting Information, Figure 6). The use of an integrated risk index such as the PRISW-1 is a precautionary approach to protecting surface water, identifying possible risk situations not highlighted by individual TERs.

The drift PRISW-1 map (Figure 5a) displays the alachlor result for the 130 portions of basins on March 30, 1996. PRISW-1 values are classified to indicate the level of risk (Table 4 in the Supporting Information) from *null* to *very high*. The index values range from 0, especially in the northern part, up to 54, in the southern Olona River. This case represents an exception and accounts for the 1.4% of the study area, while the rest of it does not exceed 27, which belongs in the class of medium risk level. The most part of the region shows medium or low risk level; null or negligible risk levels are predicted for northern basins and for the Po River while medium ones are located in the flat region.

The runoff PRISW-1 map (Figure 5b) refers to the rain event of April 2, 1996. Taking into account that areas where pedological data area lacking are not relevant for agricultural purposes (urban and mountain areas) to calculate PRISW, PEC reported in Figure 3b were extrapolated to the whole subbasin even if partial data were available. Similarly to the drift map, the null values are common in the northern part and the maximum value (PRISW-1 = 37) is again that of Olona River. Yet, in this case that value is included in the medium risk level class together with 22 others. This level of risk accounts for 43% of the runoff modeled area and is particularly present in the highland ("Alta Pianura"), in the morainic zones, in the Oltrepo Pavese where the slope increases, and also in some basins of the plain. Low to null risk levels were indicated for 56% of the area and characterized the basins of the remainder of the flat region; the null values were related to the downstream Po River segments.

Implications and Future Research

The application of the methodology is a pilot experience representing the first part of a larger project. Although the alachlor PEC and risk maps are a static image of a worst-case simulation, the main goal is to provide information on the territory referred to the watershed level, which is relevant for risk management in the aquatic environment. The information particularly focuses on investigating the driving forces of the processes under consideration and their spatial variability in order to improve knowledge about the territory and to indicate the need for site specific studies on a more detailed scale. This GIS-based methodology was combined with an ecological characterization of the receiving water bodies so that more site-specific risk indices are developed. These actions take into account the recent EU Water Framework Directive (*31*).

The advantages of the methodology consist in the following possibilities:

(i) Modeling the environmental distribution of different active ingredients applied to the same crop with the aim of evaluating the different risk level connected to their use and distribution;

(ii) Comparing and modeling different risk levels correlated to the crop management on the same area and the effectiveness of introducing Good Agricultural Practices in the use of crop protection products as suggested by EU regulatory tools (*32*);

(iii) Analyzing the processes that influence the pollution phenomenon considering different conditions in space and time, modifying scenarios of the real or potential conditions.

The use of GIS in managing the risk calculation process to get site-specific results allows to consider an high number of distributed variables and to repeat the procedure, updating the data, to get easily new result at low cost. Cartography can be load into the telematic network to share produced data in real time. The entire procedure can be managed by client-server architecture in order to permit risk evaluation and risk cartography production for a remote user with their own data set. The entire procedure will be improved in the further steps of the work, by the development of

(i) the inclusion of biological and environmental aspects such as site-specific ecological quality, vulnerability and naturalistic value of exposed ecosystems (*5*);

(ii) the application of several active ingredients with different environmental characteristics for a calibration of the procedure and a preliminary validation of exposure (*5*);

(iii) the hydrodynamic model to understand the fate of pesticides taking into account the dynamic processes of a hydrological network;

(iv) a final systematic validation of the risk assessment on pilot sites, by chemical and biological monitoring.

Supporting Information Available

Additional tables, text, figures, and references. This material is available free of charge via the Internet at http://pubs.acs.org.

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